

## In Utero Exposure to Di(*n*-butyl) Phthalate and Testicular Dysgenesis: Comparison of Fetal and Adult End Points and Their Dose Sensitivity

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**BACKGROUND:** Fetal exposure of male rats to di(*n*-butyl) phthalate (DBP) induces reproductive disorders similar to those in human testicular dysgenesis syndrome (TDS), including infertility, cryptorchidism, focal "dysgenetic areas," and Sertoli cell-only tubules in the adult testis. Humans are widely exposed to DBP, but at much lower levels than those causing adverse effects in rats.

**OBJECTIVES:** The objective of this study was to evaluate end points affected by DBP action in rats in fetal and adult life that are relevant to human TDS, and to compare their dose sensitivity.

**METHODS:** Pregnant rats were gavaged daily with corn oil (control) or with 4, 20, 100, or 500 mg/kg DBP. We examined adult end points of TDS (infertility, cryptorchidism) and indicators within the fetal testis of dysgenesis [abnormal Leydig cell (LC) aggregation, multinucleated gonocytes (MNGs)], as well as conditions that may result from these indicators in adulthood (occurrence of focal dysgenetic areas). Fetal testis weight and testicular testosterone levels were also evaluated.

**RESULTS:** The fetal end points analyzed (testicular testosterone levels, abnormal LC aggregation, occurrence of MNGs) were most sensitive to disruption by DBP, as all were significantly affected at a dose of 100 mg/kg/day DBP, with a trend toward effects occurring at 20 mg/kg/day DBP; adult end points were affected consistently only by 500 mg/kg/day DBP.

**CONCLUSIONS:** The fetal end points we evaluated can be objectively quantified and may prove helpful in evaluating the health risk of exposure to DBP and other phthalates, as well as identifying DBP-sensitive fetal events that have adult consequences/end points that are identifiable in human TDS.

**KEY WORDS:** cryptorchidism, di(*n*-butyl) phthalate, dose response, dose sensitivity, dysgenetic areas, infertility, Leydig cell aggregation, male reproductive development, multinucleated gonocytes, testicular dysgenesis syndrome. *Environ Health Perspect* 115(suppl 1):55–61 (2007). doi:10.1289/ehp.9366 available via <http://dx.doi.org/> [Online 8 June 2007]

Disorders of male reproductive health are common and perhaps increasing in the Western world (Sharpe and Skakkebaek 2003; Toppari et al. 1996). These include cryptorchidism and hypospadias, which present at birth, and low sperm counts and testicular germ cell cancer, which manifest in young adulthood. Evidence suggests that these disorders may have a common origin in fetal life (Sharpe and Skakkebaek 2003; Skakkebaek et al. 2001). Based on this evidence, these disorders have been hypothesized to constitute a testicular dysgenesis syndrome (TDS), which results from abnormal development of the testis and consequent malfunction of the Leydig cells (LC) and/or Sertoli cells (SC) during male sexual differentiation (Sharpe and Skakkebaek 2003; Skakkebaek et al. 2001).

Certain phthalate esters such as di(*n*-butyl) phthalate (DBP) have been shown in rats to interfere with normal development of the testis and reproductive tract when exposure occurs during gestation (Barlow and Foster 2003; Ema et al. 1998, 2000; Fisher et al. 2003; Gray et al. 1999; Mylchreest et al. 1998, 1999, 2000), resulting in postnatal downstream disorders that are similar to those reported in human TDS, that is, cryptorchidism, hypospadias, impaired spermatogenesis, and reduced male fertility. Fetal exposure of male rats to DBP therefore has

potential as an animal model in which to explore the events in the fetal testis that give rise to TDS (Fisher et al. 2003). In searching for such mechanisms, various phthalate-induced changes in the fetal testis have been reported, namely the occurrence of multinucleated gonocytes (MNGs), abnormal LC aggregation, decreased testicular testosterone levels, and decreased insulin-like factor 3 mRNA/protein expression (Fisher et al. 2003; Mahood et al. 2005; McKinnell et al. 2005; Parks et al. 2000; Shultz et al. 2001; Wilson et al. 2004). We have also shown that *in utero* exposure of rats to 500 mg/kg DBP causes postnatal development of focal areas of dysgenesis, comprising malformed seminiferous cords/tubules with intracordal/intratubular LC and immature SC, within otherwise normal testes that exhibit complete spermatogenesis (Fisher et al. 2003; Mahood et al. 2005, 2006); similar changes have been reported in TDS (testicular cancer) patients (Hoei-Hansen et al. 2003; Sharpe 2006).

The similarity between DBP-induced TDS-like disorders in rats and the relatively high prevalence of TDS disorders in humans has raised the question of whether DBP, or other phthalates, might contribute causally to human TDS. This possibility arises because of the widespread exposure of humans to DBP and other phthalates (Hauser et al.

2004; Rudel et al. 2003; Schettler 2006), although there is only one piece of published evidence to indicate that phthalate exposure during pregnancy might cause changes in the male offspring that are analogous to those reported in rats (Swan et al. 2005). An important issue in this debate is that the lowest doses (50–100 mg/kg/day) of DBP, or other phthalates, that have been shown to cause abnormal changes in rats, namely reduced anogenital distance (AGD) and nipple retention (Mylchreest et al. 2000; Zhang et al. 2004), are considerably higher than reported exposure levels in the human population (Hauser and Calafat 2005; Koch et al. 2003; Silva et al. 2004). Additionally, most of the end points evaluated in rat studies, such as AGD (Barlow and Foster 2003; Barlow et al. 2004; Carruthers and Foster 2005; Ema et al. 1998, 2000; Mylchreest et al. 1998, 1999, 2000; Zhang et al. 2004) and nipple retention (Barlow and Foster 2003; Barlow et al. 2004; Carruthers and Foster 2005; Mylchreest et al. 1999, 2000) are difficult to relate directly to TDS in humans.

Therefore, the aim of the present study was to evaluate end points in our *in vivo* rat model of TDS that can be related more obviously to similar TDS disorders in humans, and to explore the DBP dose sensitivity of these end points. It was also important that all of these end points could be quantifiable to enable objective measurement. The end points that we chose included actual end points of TDS (infertility, cryptorchidism), together with indicators within the testis of dysgenesis (occurrence of dysgenetic areas, MNGs, and abnormal LC aggregation). Other fetal end points, such as testis weight and testicular testosterone levels, were also evaluated. Where relevant, these end points were investigated in

This article is part of the monograph "Endocrine Disruptors—Exposure Assessment, Novel End Points, and Low-Dose and Mixture Effects."

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We thank M. Fisk for expert animal husbandry.

This work was supported in part by grants QLK4-199-01422 and QLK4-CT-200-00603 from the European Union.

The authors declare they have no competing financial interests.

Received 22 May 2006; accepted 8 February 2007.

both fetal and postnatal life, and four different doses of DBP (4, 20, 100, and 500 mg/kg/day) were used. Our results show that it is the fetal end points that are the most sensitive to DBP action.

## Materials and Methods

**Animals, treatments, sample collection, and processing.** Wistar rats were maintained in our own animal facility according to UK Home Office guidelines [Animal (Scientific Procedures) Act 1986], and were fed a soy-free breeding diet (SDS, Dundee, Scotland). Time-mated females [day of vaginal plug = gestation day (GD) 0.5] were treated from GD13.5 to either GD20.5 (fetal samples) or GD21.5 (postnatal tissue) with either 0 (control), 4, 20, 100, or 500 mg/kg DBP (Sigma, Dorset, UK) in 1 mL/kg corn oil administered daily by oral gavage. The DBP was 99% pure according to the supplier. Corn oil was obtained from a supermarket and was used as obtained.

**Fetal samples.** Control and DBP-treated pregnant dams were killed by inhalation of carbon dioxide followed by cervical dislocation on GD21.5. Fetuses were removed, decapitated, and placed in ice-cold phosphate-buffered saline (PBS; Sigma). Testes were removed via microdissection, fixed for 1 hr in Bouins fixative, and then transferred to 70% ethanol. Fixed testes were weighed and then processed into paraffin wax using standard methods. One or more male fetuses from each litter were subsequently used for the quantitative and immunohistochemical studies described below; selection of fetuses for further study was random.

**Adult samples.** Male rats 90 days of age were killed by inhalation of CO<sub>2</sub> followed by cervical dislocation. Testes were carefully inspected for normality of the epididymis and vas deferens and then removed, weighed, fixed for 5–6 hr in Bouins fixative, and then transferred into 70% ethanol. Testes were halved after approximately 3 hr fixation to aid penetration of the fixative. Testes were then further cut into four to eight blocks, depending on size, and embedded in paraffin as described above. At necropsy, testicular position was classified as high abdominal (at level of the kidney), mid-abdominal, inguinal, or scrotal, which enabled classification of testes into cryptorchid or scrotal groups. In controls, all testes were scrotal in position. Individual testes from adult animals that exhibited any gross epididymal lesions were excluded from histologic analysis to avoid possible confounding effects of this change on testicular morphology (Barlow and Foster 2003; Mylchreest et al. 1998); this applied to two testes from a total of 13 males. Before dissection, the adult male rats underwent a fertility test. This involved each male rat

being housed singly for 1 week with a female rat of proven fertility. Males were classed as fertile if offspring were produced.

For the studies above, animals were treated humanely and with regard for alleviation of suffering. All studies were performed according to the Animal (Scientific Procedures) Act (1986) under Project Licence approval by the UK Home Office.

**Testicular testosterone analysis.** Testicular testosterone levels were measured by radioimmunoassay, as described previously (Fisher et al. 2003), on GD21.5 in individual testes from males from four to six litters per treatment group. After dissection, testes were snap frozen on dry ice and stored at –70°C before analysis. Testes were defrosted and homogenized individually in 0.5 mL PBS; an aliquot of this solution was then extracted with 2 mL diethyl ether, shaken for 5 min, and then placed in a bath of methanol cooled with dry ice. The nonaqueous portion of the extract was then decanted, dried overnight in a fume hood, and reconstituted in assay buffer. The limit of detection of the assay was 40 pg/testis.

**Immunohistochemistry.** Specific proteins were detected by immunohistochemistry using methods that have been described previously (Fisher et al. 2003). Sections (5 µm) were mounted onto coated slides (BDH Chemicals, Poole, UK), dewaxed, and rehydrated. Slides were incubated in 3% (vol/vol) hydrogen peroxide in methanol to block endogenous peroxidase activity, then washed in Tris-buffered saline (TBS: 0.05 M Tris, 0.85% NaCl, pH 7.4 at room temperature). Non-specific binding sites were blocked with an appropriate normal serum diluted 1:5 in TBS containing 5% bovine serum albumin (Sigma). Sections were incubated with the primary antibodies 3β-hydroxysteroid dehydrogenase (3β-HSD; 1:4000; gift from I. Mason, Edinburgh, UK) or smooth muscle actin (SMA; 1:2000; Sigma) overnight at 4°C.

The next morning, slides were incubated 30 min with the appropriate secondary antibody conjugated to biotin, at a dilution of 1:500 (rabbit anti-mouse or swine anti-rabbit; DAKO, Cambridgeshire, UK). The biotinylated antibody was linked to horseradish peroxidase (HRP) by 30-min incubation with avidin-biotin–HRP complex (ABC–HRP; DAKO). Antibody localization was determined by application of diaminobenzidine (liquid DAB<sup>+</sup>; DAKO) until staining in control sections was optimal; the reaction was stopped by immersing slides in distilled water. Slides were counterstained with hematoxylin, dehydrated, and mounted using Pertex mounting media (Cell Path; Hemel Hempstead, UK).

**Analysis of MNGs.** We identified MNGs using toluidine blue staining of GD21.5 testis sections from one or more testes from animals selected at random from five to nine litters per

treatment group. Slides were dewaxed and rehydrated as for immunohistochemistry. The toluidine blue stain (BDH Chemicals) was filtered and applied to slides at a 50% dilution with distilled water. Once staining was optimal the slides were immersed in distilled water and then dehydrated and mounted. To analyze the occurrence of MNGs, we used an Olympus BH-2 microscope (Olympus Optical, London, UK) fitted with a Prior automatic stage (Prior Scientific Instruments Ltd, Cambridge, UK). One complete testis cross-section from each animal was analyzed, and the percentage of seminiferous cord cross-sections that contained one or more MNGs was recorded. We did not record the number of MNGs per cord or the number of nuclei per MNG. The analyzer was blinded with respect to the treatment status of animals.

**LC cluster analysis.** LC aggregation was objectively quantified using previously described methods (Mahood et al. 2005). Testes from one or more fetuses per litter at GD21.5, from five to six litters per treatment group, were randomly selected and serially sectioned. DBP treatment results in unequal distribution of LC throughout the testis due to their abnormal migration/aggregation into more central regions (Mahood et al. 2005). To avoid sampling errors because of this unequal distribution, we selected three sections per testis that corresponded to uniform intervals through the serially sectioned testis (25, 50, and 75% intervals, which were thus 20–30 sections apart from each other); these sections were immunostained for 3β-HSD as described above.

Quantification of LC clustering in these sections was undertaken using Image-Pro Plus 4.5.1 software and equipment (Media Cybernetics UK, Wokingham, Berkshire, UK). Specimens immunostained for 3β-HSD were of sufficient homogeneity, high contrast, and low background to allow computer-assisted thresholding and subsequent computer-assisted counting of LC (3β-HSD–immunopositive) clusters and determination of LC cluster area. Digital images of complete testis sections were captured at ×40 magnification. The software was used to trace around each section, creating an area of interest; the area of each section could then be calculated. Computer-assisted thresholding was then used to identify and analyze clusters of 3β-HSD–immunopositive cells, generating data on cluster number and the proportion of each section occupied by LC clusters, as described in detail elsewhere (Mahood et al. 2005). To take account of the decrease in LC size in DBP-exposed males (Mahood et al. 2005), each LC cluster was expressed as a percentage of the total LC cluster area in each animal. Clusters were then arbitrarily assigned to one of three groups: small clusters, accounting for ≤ 5% of the total

LC cluster area per testis; medium clusters, accounting for 5.1–14.9%; and large clusters, which individually accounted for  $\geq 15\%$  of the total LC cluster area per testis.

**Incidence of dysgenesis.** We assessed the incidence of dysgenesis in each of the treatment groups in adult male rats by visual analysis of SMA immunostained sections from four to eight blocks per testis for each animal. Sections were checked for the occurrence of Sertoli cell-only (SCO) tubules and focal dysgenetic areas in each testis (cryptorchid and scrotal). Testes were obtained from one or more animals from at least five separate litters from each treatment group.

**Image capture.** Images were examined and photographed using a Provis microscope (Olympus Optical, London, UK) fitted with a Kodak DCS330 digital camera (Eastman Kodak, Rochester, NY, USA). Images were compiled using Photoshop 7.0 (Adobe Systems Inc., Mountain View, CA, USA).

**Statistical analysis.** We used data for each of the fetal end points and for adult testis weight to derive a mean value for each litter; litter means  $\pm$  SE were then computed and used for statistical analyses. Data were analyzed by one-way analysis of variance followed by the Bonferroni post-test. The incidence of infertility, cryptorchidism, SCO tubules, and focal

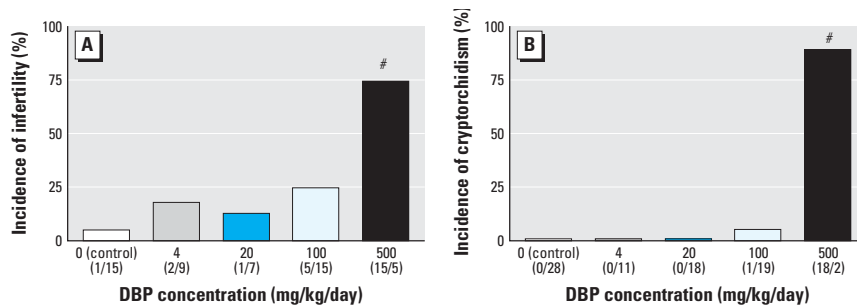
dysgenetic areas in adult animals was analyzed using Fisher's exact test. For the latter datasets, we analyzed both data from individual animals and litter means, in view of the wide variation (0–100%) in incidence between different litters for some of the measured end points. Data for LC cluster number per testis were log transformed before statistical analysis to normalize variances. All analyses were performed using GraphPad Prism (version 4; GraphPad Software Inc., San Diego, CA, USA).

## Results

**Infertility, cryptorchidism, testis weight, and testicular testosterone.** In adult animals prenatally exposed to 500 mg/kg DBP, we found a 75% infertility rate, dropping to 33% in the 100-mg/kg group and to 14% in the 20-mg/kg group. However, only the 500 mg/kg DBP treatment group had significantly reduced fertility compared with the controls when data were analyzed for individual animals (Figure 1A) or when analyzed using incidence of infertility per litter ( $p = 0.03$ ). The 500-mg/kg DBP treatment group was also the only treatment group to have a significantly elevated incidence of cryptorchidism compared with the control group, with 90% of animals exhibiting either unilateral or bilateral cryptorchidism compared with no animals in

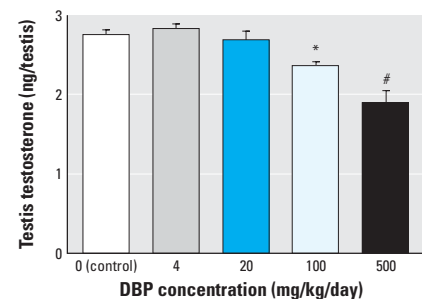
the control group (Figure 1B); this difference was equally evident when data were analyzed as incidence of cryptorchidism per litter ( $p = 0.005$ ). In all other treatment groups, we found only one case of cryptorchidism (unilateral), and this was in the group exposed to 100 mg/kg DBP. Exposure to 500 mg/kg DBP during gestation resulted in a significant decrease in testis weight compared with control animals both at GD21.5 and in adulthood (Figure 2A,B), although the reduction in adulthood was attributable solely to the high incidence of cryptorchid testes (Figure 2C). Animals exposed to 4, 20, or 100 mg/kg DBP did not show any significant change in testis weight at either GD21.5 (Figure 2A) or in adulthood (Figure 2B). Testicular testosterone levels in GD21.5 animals were significantly decreased in both the 500- and 100-mg/kg DBP treatment groups compared with control values (Figure 3). We found no significant effect of DBP treatment on fetal body weight or on litter size (data not shown).

**Occurrence of MNGs at GD21.5.** All treatment groups, including the control group, had MNGs within the seminiferous cords at GD21.5 (Figure 4A,B). Prenatal exposure to either 500 or 100 mg/kg DBP resulted in a significant increase in the occurrence of MNGs compared with controls; also,



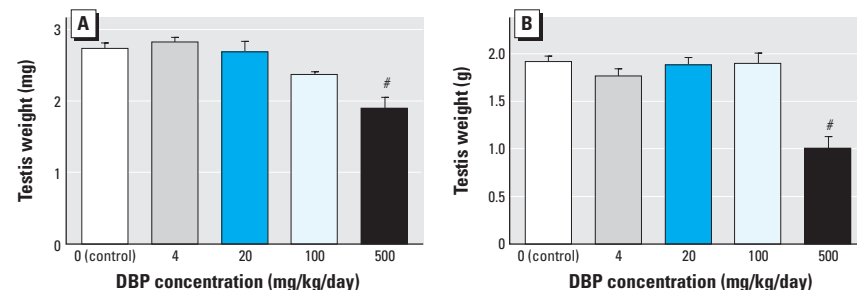
**Figure 1.** Incidence of infertility (A) and cryptorchidism (B) in adult male rats exposed *in utero* (GD13.5–GD21.5) to corn oil (control) or DBP. Values shown in parentheses are the number of infertile/fertile animals (A) and the number of animals with either unilateral or bilateral cryptorchid/normal testis position (B). Animals were derived from five to seven litters except for the 20 mg/kg DBP group, which was derived from three litters.

\* $p < 0.0001$  compared with control.



**Figure 3.** Testicular testosterone levels at GD21.5 in animals exposed *in utero* (GD13.5–GD20.5) to corn oil (control) or DBP. Testosterone values are litter mean  $\pm$  SE for four to six litters per group.

\* $p < 0.05$ . # $p < 0.001$  compared with control.



**Figure 2.** Testis weights at GD21.5 (A) or in adulthood (B) in male rats exposed *in utero* (GD13.5–GD21.5) to corn oil (control) or DBP. (C) Breakdown comparison of cryptorchid and scrotal testis weights from adult male rats exposed *in utero* to 500 mg/kg DBP. Values are litter mean  $\pm$  SE for 4–16 litters per treatment group (based on 28–98 animals/group).

\* $p < 0.001$  compared with respective control s. ## $p < 0.0001$  compared with cryptorchid testis weights.



we found an increase in the occurrence of MNGs in animals exposed to 20 mg/kg DBP (Figure 4C), although it was not statistically significant.

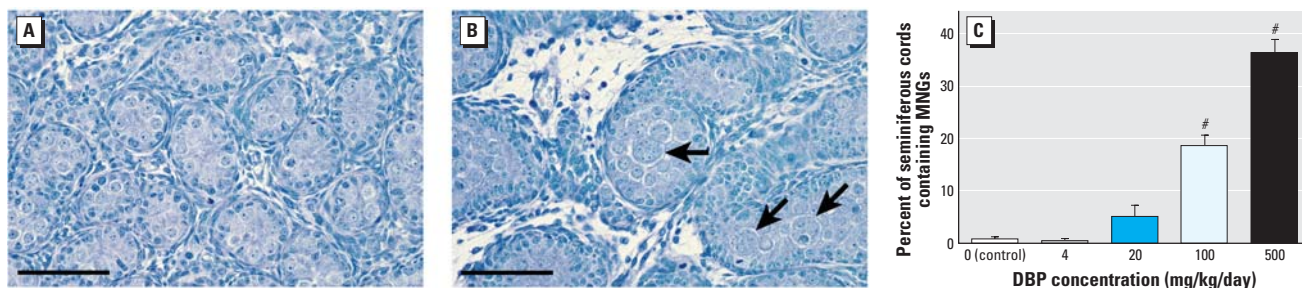
**LC clustering/aggregation at GD21.5.** Changes in LC distribution were obvious in both the 100- and 500-mg/kg DBP treatment groups (Figure 5D,E) compared with controls (Figure 5A); these changes were most pronounced in the highest dose group, with large LC clusters being evident in the center of the testes (Figure 5E). LC distribution in testis sections from the 4- and 20-mg/kg treatment groups were not obviously different from controls (Figures 5B,C). Objective analysis of LC aggregation revealed a significant decrease in total LC cluster number per testis section in animals exposed to either 100 or 500 mg/kg DBP (Figure 6A). This pattern was similarly reflected in the data for the percentage of total cluster area accounted for by small clusters (Figure 6B). LC clusters of medium size were evident in all treatment groups but were only significantly increased above control values in the 100-mg/kg DBP group (Figure 6C). The occurrence of large LC clusters was evident only in groups of males exposed to 20, 100, or 500 mg/kg DBP (Figure 6D). However, compared with control animals, the number of large LC clusters was significantly increased only in the 500-mg/kg group, in which approximately 40% of the total LC cluster area was accounted for by clusters of this size (Figure 6D).

#### *Incidence of focal dysgenesis in adulthood.*

Visual analysis of SMA-immunostained testis sections from each treatment group revealed that all cryptorchid testes examined from the 500- and 100-mg/kg DBP groups had SCO tubules present (Table 1). Of the 11 cryptorchid testes examined in the 500-mg/kg DBP group, 7 had one or more focal dysgenetic areas. In testes from control animals, we found neither SCO tubules nor areas of focal dysgenesis (Figure 7A). SCO tubules were found in all DBP-exposed groups except the 4 mg/kg treatment group (Table 1). However, not all testes examined in each group had SCO tubules present. Interestingly, we found that a similar number of testes in the two highest dose groups (500 and 100 mg/kg) had SCO tubules (~ 66% of testes examined); this incidence was statistically significant when data were analyzed for individual animals (Table 1) or based on the incidence per litter (i.e., any animal per litter exhibiting SCO tubules;  $p = 0.04$ ). Focal dysgenetic areas were detected only in the 500- and 100-mg/kg DBP animals (Figure 7B,C); 55% of testes examined in the 500-mg/kg group and approximately 33% in the 100-mg/kg dose group had areas of dysgenesis present. However, only the incidence of dysgenetic areas in the 500-mg/kg DBP group achieved statistical significance when analyzed for individual animals (Table 1); this was of borderline significance when evaluated per litter ( $p \sim 0.05$ ).

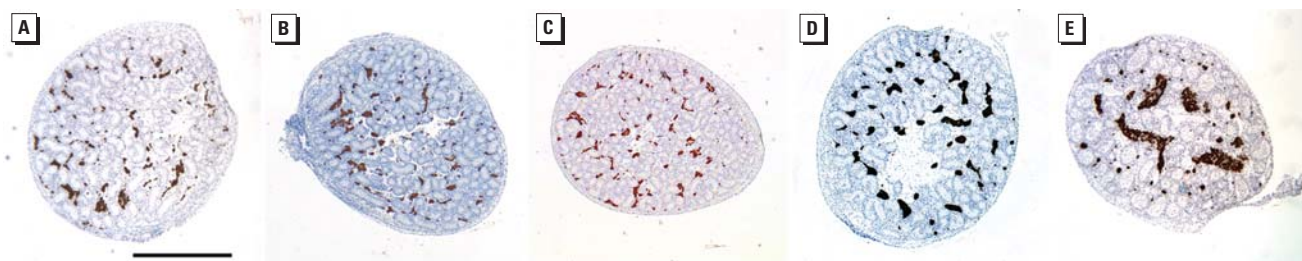
## Discussion

We and others have shown that fetal exposure of male rats to DBP, or certain other phthalates, results in a high incidence of disorders such as cryptorchidism, hypospadias, and infertility (Barlow and Foster 2003; Ema et al. 1998, 2000; Fisher et al. 2003; Gray et al. 1999; Mylchreest et al. 1998, 1999, 2000). These disorders are collectively similar to those reported in human TDS patients, and as such, the focus of the present study was to investigate DBP-induced changes in fetal life and in adulthood that were considered relevant to TDS. To date, studies investigating the effects of DBP on male reproductive development have used end points that may be relevant to human TDS (occurrence of cryptorchidism, decreased fertility), or that are not relevant (nipple retention in male rats), or that are presently of uncertain relevance (decreased AGD) (Barlow and Foster 2003; Barlow et al. 2004; Carruthers and Foster 2005; Ema et al. 1998, 2000; Mylchreest et al. 1998, 1999, 2000; Zhang et al. 2004). Similarly, changes in gene and protein expression after DBP exposure, which have been found to be more sensitive to the effects of DBP (Lehmann et al. 2004), cannot be related directly to TDS other than when the changes relate to expression of genes involved in LC hormone production. These include reduction in the expression of genes and proteins involved in cholesterol transport and steroidogenesis leading to a concomitant



**Figure 4.** Occurrence of MNGs (arrows) in representative photomicrographs of testicular sections collected on GD21.5 from rats exposed *in utero* (GD13.5–GD20.5) to corn oil (control; A) or 500 mg/kg/day DBP (B). Tissues were stained with toluidine blue; bar = 50  $\mu$ m. (C) Percentage of seminiferous cords exhibiting MNGs on GD21.5 in animals exposed *in utero* to DBP (mg/kg/day); values are litter mean  $\pm$  SE for five to nine litters per treatment group.

\* $p < 0.001$  compared with control.



**Figure 5.** Representative photomicrographs illustrating the change in distribution of Leydig cells (3 $\beta$ -HSD-positive, brown) in the testes of GD21.5 animals exposed *in utero* (GD13.5–GD20.5) to corn oil (control; A) or DBP doses of (B) 4, (C) 20, (D) 100, or (E) 500 mg/kg/day. Tissues were counterstained with hematoxylin; bar = 0.5 mm.

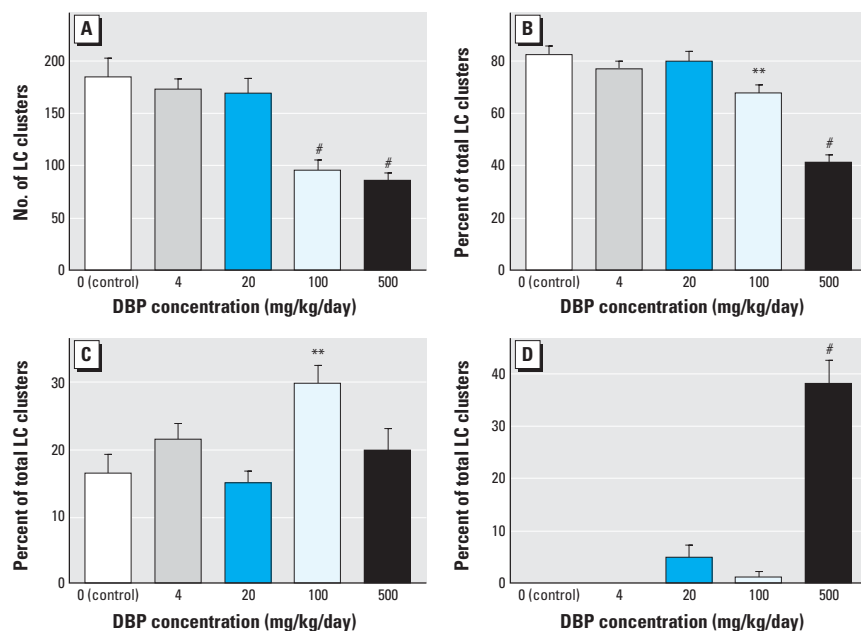
decrease in fetal testicular testosterone levels, as seen in the present study at doses  $\geq 100$  mg/kg DBP, and as has also been reported to occur at doses as low as 50 mg/kg/day DBP (Barlow

et al. 2003; Lehmann et al. 2004; Shultz et al. 2001). The reduction in AGD and the retention of nipples in male rats exposed during gestation to DBP are presumed to be effects of

DBP that are secondary to decreased testicular testosterone production (Fisher et al. 2003; Mahood et al. 2005; Mylchreest et al. 2002; Parks et al. 2000; Shultz et al. 2001). For this reason, and because dose–response studies using these end points are already well reported (Mylchreest et al. 1998, 1999, 2000; Zhang et al. 2004), we did not measure AGD or nipple retention in the present study. Instead, our aim was to evaluate objectively quantifiable end points that we considered of direct relevance to either the manifestation (adult end points) or origins (fetal end points) of TDS.

Our results indicate that it is the fetal end points analyzed (testicular testosterone levels, abnormal LC aggregation, occurrence of MNGs) that appear to be most sensitive to disruption by DBP. All three of these end points were affected significantly by a DBP dose of 100 mg/kg/day; it also appears that a dose of 20 mg/kg/day probably also has an effect on some of these parameters, although this did not achieve statistical significance. Only exposure to 500 mg/kg/day DBP resulted in a significant decrease in testis weight in fetal and adult life, but even in the latter, the observed decrease was secondary to the occurrence of cryptorchidism. Thus, segregation of adult testis weights into scrotal and cryptorchid testes in the 500-mg/kg dose group showed that the decrease in mean testis weights seen at this dose was due completely to the small size of the cryptorchid testes in these animals. In this regard, it is also notable that in the 500-mg/kg/day DBP treatment group, adult scrotal testes, which were normal in weight and largely normal for gross morphology and completeness of spermatogenesis, clearly exhibited focal abnormalities (focal dysgenesis, SCO tubules). The other postnatal end points investigated in this study (incidence of infertility and cryptorchidism) were significantly increased above control levels only in the 500 mg/kg/day group, indicating that these are insensitive end points to use when investigating the effects of lower dose DBP exposure in fetal life.

We have recently proposed a model to explain the origin of dysgenetic areas and



**Figure 6.** Number and distribution of LC clusters in testes collected on GD21.5 from male rats exposed *in utero* (GD13.5–GD20.5) to corn oil (control) or to DBP. The number of LC clusters per testis section (A) and the percentage occurrence of small (B), medium (C), and large (D) LC clusters are shown for each treatment group. Values shown are litter mean  $\pm$  SE for five to six litters per treatment group. Small clusters account for  $\leq 5\%$  of the total LC cluster area per testis, medium clusters for 5.1–14.9%, and large clusters for  $\geq 15\%$ .

\*\* $p < 0.01$ . # $p < 0.001$  compared with respective control values.

**Table 1.** Incidence of SCO tubules and dysgenetic areas in the testes of adult male rats exposed *in utero* (GD13.5–GD21.5) to corn oil (control) or to varying doses of DBP (mg/kg/day).

DBP (mg/kg/day)	Cryptorchid testes		Scrotal testes		Dysgenetic areas
	No. analyzed	SCO tubules present	No. analyzed	SCO tubules present	
0 (control)	0	—	9	No (0/9)	No (0/9)
500	11	Yes (11/11) <sup>a</sup>	9	Yes (6/9) <sup>b</sup>	Yes (5/9) <sup>c</sup>
100	1	Yes	12	Yes (8/12) <sup>d</sup>	Yes (4/12)
20	0	—	5	Yes (1/5)	No (0/5)
4	0	—	11	No (0/11)	No (0/11)

Numbers in parentheses indicate the number of testes that exhibited either SCO tubules or dysgenetic areas of the total number of testes analyzed. Each scrotal and cryptorchid testis analyzed was from a different animal and represented at least three different litters; seven of the analyzed cryptorchid testes were from unilaterally cryptorchid animals for which the corresponding scrotal testis was also analyzed.

<sup>a</sup>Seven of these 11 testes had dysgenetic areas. <sup>b</sup> $p = 0.009$ . <sup>c</sup> $p = 0.029$ . <sup>d</sup> $p = 0.005$ .



**Figure 7.** Photomicrographs of scrotal testes from adult rats exposed *in utero* to (A) corn oil (control), or to DBP doses of (B) 500 or (C) 100 mg/kg/day DBP. Although the majority of each cross-section of scrotal testes in DBP-exposed animals had normal seminiferous tubules, some animals exhibited one or more focal dysgenetic areas comprising malformed seminiferous tubules as visualized by immunostaining for SMA to identify peritubular myoid cells. Note the occurrence of unusually large numbers of LCs surrounding the malformed tubules (B,C). Bar = 100  $\mu$ m.



SCO tubules formed in the testes of DBP-exposed rats (Mahood et al. 2005, 2006). This model hypothesizes that DBP-induced abnormal migration and aggregation of LCs within the fetal testis "traps" isolated SC, gonocytes, and presumably peritubular myoid cells within them (Fisher et al. 2003). Postnatally, after cessation of DBP treatment, these non-segregated clusters of cells then try to form seminiferous cords; we propose that this process results in the formation of dysgenetic areas focally within the testes, surrounded by otherwise normal tissue with complete spermatogenesis (in scrotal testes). The results from the present dose-response study support our proposed model because dysgenetic areas were seen in adulthood following fetal exposure to either 100 or 500 mg/kg/day DBP, and it is at these doses that abnormal LC aggregation was induced in fetal life. The dose-dependent nature of fetal LC aggregation was grossly evident in the testicular images immunostained for 3 $\beta$ -HSD (Figure 5) as well as after objective quantification using image analysis. An increase in the occurrence of MNGs was also observed at GD21.5 in the 100 mg/kg and the 500 mg/kg DBP dose groups (Figure 4), although it is still unknown if these abnormal germ cells contribute to the dysgenetic phenotype of the adult DBP-exposed animals and what their precise relevance, if any, is to human TDS (Ferrara et al. 2006). The cause of MNGs is unknown, but it is not due to abnormal/incomplete cell division because it occurs at ages when the germ cells are quiescent and not dividing (GD19.5–GD21.5); we have confirmed this using a number of different proliferation markers (Ferrara et al. 2006). It seems likely that the mechanisms underlying the induction of MNGs are separate from those involved in the formation of focal dysgenetic areas, because MNGs occur throughout the fetal testis and are equally evident in normally formed seminiferous cords as in areas destined to form dysgenetic areas (Fisher et al. 2003).

A dose-response study investigating changes in gene and protein expression in the fetal testis established a dose of DBP of 50 mg/kg/day as the lowest observed adverse effect level (LOAEL), and 10 mg/kg/day as the no observed adverse effect level (NOAEL) (Lehmann et al. 2004). In the present study, we did not use a dose of 50 mg/kg/day, but we have established a LOAEL of 100 mg/kg/day for fetal end points and a NOAEL of 20 mg/kg/day, although there does appear to be a clear trend towards effects on the occurrence of MNGs at a dose of 20 mg/kg/day DBP; this may have reached statistical significance if more litters had been studied. Taken together, the available data suggests that doses of 20–50 mg/kg/day DBP may cause changes in fetal rat testis development that are of likely

relevance to the postulated origins of human TDS disorders (i.e., disruption of fetal testis organization/function). These levels of exposure are still much higher than the suggested environmental exposure levels of the human population (Hauser and Calafat 2005; Koch et al. 2003; Silva et al. 2004). Although this contrast suggests, at face value, that the human male fetus is at low risk from exposure to DBP alone, it is important to consider the combined risk of exposure to DBP and other phthalates that can exert similar effects (e.g., diethylhexyl phthalate), or indeed other endocrine-disrupting chemicals. Recent studies have reported that pairs of phthalates (Borch et al. 2004) or combination of a phthalate with another anti-androgen with a different mode of action (Hotchkiss et al. 2004) produce dose-additive effects on male reproductive malformations in rats. Therefore, studies investigating the dose-response relationships and dose-additive effects of phthalates and other endocrine-disrupting chemicals on male reproductive development need to be a focus of future research in order to fully evaluate the risk of human exposure.

We hope that the new fetal end points described in the present study (occurrence of MNGs and abnormal LC aggregation), which can be objectively quantified, will prove a useful resource in future studies. It can be argued that the fetal occurrence of MNGs induced by DBP is irrelevant because it has no known direct consequences for the fetus (which does not reproduce) or the adult. However, because MNGs may reflect altered SC function in fetal life (Ferrara et al. 2006), and it is the number (and function) of the SC that predetermines sperm count and fertility in adulthood, MNGs may provide a sensitive (indirect) measure of such effects. Similarly, the occurrence of abnormal LC aggregation in the fetal testis is potentially of direct evidence to adult testis function, as our evidence suggests that it is an important cause of SCO tubules and focal dysgenetic areas in the testis in adulthood (Mahood et al. 2006). As the latter features are clearly relevant to human TDS (Hoei-Hansen et al. 2003; Sharpe 2006), it would be sensible to concentrate on such end points or to identify other fetal end points that have proven consequences that are detectable in adulthood.

## REFERENCES

- Animal (Scientific Procedures) Act. 1986. London: Home Office. Available: <http://www.archive.official-documents.co.uk/document/hoc/321/321-xa.htm> [accessed 5 February 2007].
- Barlow NJ, Foster PM. 2003. Pathogenesis of male reproductive tract lesions from gestation through adulthood following in utero exposure to di(*n*-butyl) phthalate. *Toxicol Pathol* 31:397–410; doi:10.1080/0192623030909723.
- Barlow NJ, McIntyre BS, Foster PM. 2004. Male reproductive tract lesions at 6, 12, and 18 months of age following in utero exposure to di(*n*-butyl) phthalate. *Toxicol Pathol* 32:79–90; doi:10.1080/01926230490265894.
- Barlow NJ, Phillips SL, Wallace DG, Sar M, Gaido KW, Foster PM. 2003. Quantitative changes in gene expression in fetal rat testes following exposure to di(*n*-butyl) phthalate. *Toxicol Sci* 73:431–441; doi:10.1093/toxsci/kfg087.
- Borch J, Ladefoged O, Hass U, Vinggaard AM. 2004. Steroidogenesis in fetal male rats is reduced by DEHP and DINP, but endocrine effects of DEHP are not modulated by DEHA in fetal, prepubertal and adult male rats. *Reprod Toxicol* 18:53–61; doi:10.1016/j.reprotox.2003.10.011.
- Carruthers CM, Foster PM. 2005. Critical window of male reproductive tract development in rats following gestational exposure to di-*n*-butyl phthalate. *Birth Defects Res B Dev Reprod Toxicol* 74:277–285; doi:10.1002/bdrb.20050.
- Ema M, Miyawaki E, Kawashima K. 1998. Further evaluation of developmental toxicity of di-*n*-butyl phthalate following administration during late pregnancy in rats. *Toxicol Lett* 98:87–93; doi:10.1016/S0378-4274(98)00107-6.
- Ema M, Miyawaki E, Kawashima K. 2000. Critical period for adverse effects on development of reproductive system in male offspring of rats given di-*n*-butyl phthalate during late pregnancy. *Toxicol Lett* 111:271–278; doi:10.1016/S0378-4274(99)00192-7.
- Ferrara D, Hallmark N, Scott HS, Brown R, McKinnell C, Mahood IK et al. 2006. Acute and long-term effects of in utero exposure of rats to di(*n*-butyl) phthalate on testicular germ cell development and proliferation. *Endocrinology* 147:5352–5362; doi:10.1210/en.2006.0527.
- Fisher JS, Macpherson S, Marchetti N, Sharpe RM. 2003. Human 'testicular dysgenesis syndrome': a possible model using in-utero exposure of the rat to dibutyl phthalate. *Hum Reprod* 18:1383–94; doi:10.1093/humrep/deg273.
- Gray LE Jr, Wolf C, Lambright C, Mann P, Price M, Cooper RL, et al. 1999. Administration of potentially antiandrogenic pesticides (procymidone, linuron, iprodione, chlozolinate, *p,p'*-DDE, and ketoconazole) and toxic substances (dibutyl- and diethylhexyl phthalate, PCB 169, and ethane dimethane sulphonate) during sexual differentiation produces diverse profiles of reproductive malformations in the male rat. *Toxicol Ind Health* 15:94–118; doi:10.1191/074823399678846664.
- Hauser R, Calafat AM. 2005. Phthalates and human health. *Occup Environ Med* 62:806–818; doi:10.1136/oem.2004.017590.
- Hauser R, Duty S, Godfrey-Bailey L, Calafat AM. 2004. Medications as a source of human exposure to phthalates. *Environ Health Perspect* 112:751–753.
- Hoei-Hansen CE, Holm M, Rajpert-De Meyts E, Skakkebaek NE. 2003. Histological evidence of testicular dysgenesis in contralateral biopsies from 218 patients with testicular germ cell cancer. *J Pathol* 200:370–374; doi:10.1002/path.1372.
- Hotchkiss AK, Parks-Saldutti LG, Ostby JS, Lambright C, Furr J, Vandenberg JG et al. 2004. A mixture of the "anti-androgens" linuron and butyl benzyl phthalate alters sexual differentiation of the male rat in a cumulative fashion. *Biol Reprod* 71:1852–1861; doi:10.1095/biolreprod.104.031674.
- Koch HM, Drexler H, Angerer J. 2003. An estimation of the daily intake of di(2-ethylhexyl)phthalate (DEHP) and other phthalates in the general population. *Int J Hyg Environ Health* 206:77–83; doi:10.1078/1438-4639-00205.
- Lehmann KP, Phillips S, Sar M, Foster PM, Gaido KW. 2004. Dose-dependent alterations in gene expression and testosterone synthesis in the fetal testes of male rats exposed to di(*n*-butyl) phthalate. *Toxicol Sci* 81:60–68; doi:10.1093/toxsci/kfh169.
- Mahood IK, Hallmark N, McKinnell C, Walker M, Fisher JS, Sharpe RM. 2005. Abnormal Leydig cell aggregation in the fetal testis of rats exposed to di(*n*-butyl) phthalate and its possible role in testicular dysgenesis. *Endocrinology* 146:613–623; doi:10.1210/en.2004-0671.
- Mahood IK, McKinnell C, Walker M, Hallmark N, Scott H, Fisher JS, et al. 2006. Cellular origins of testicular dysgenesis in rats exposed in utero to di(*n*-butyl) phthalate. *Int J Androl* 29:148–154; doi:10.1111/j.1365-2605.2005.00574.x.
- McKinnell C, Sharpe RM, Mahood K, Hallmark N, Scott H, Ivell R, et al. 2005. Expression of insulin-like factor 3 protein in the rat testis during fetal and postnatal development and in relation to cryptorchidism induced by in utero exposure to di(*n*-butyl) phthalate. *Endocrinology* 146:4536–4544; doi:10.1210/en.2005-0676.
- Mylchreest E, Cattley RC, Foster PM. 1998. Male reproductive tract malformations in rats following gestational and lactational exposure to di(*n*-butyl) phthalate: an antiandrogenic mechanism? *Toxicol Sci* 43:47–60.
- Mylchreest E, Sar M, Cattley RC, Foster PM. 1999. Disruption of androgen-regulated male reproductive development by di(*n*-butyl) phthalate during late gestation in rats is different from

- flutamide. *Toxicol Appl Pharmacol* 156:81–95; doi:10.1006/taap.1999.8643.
- Mylchreest E, Sar M, Wallace DG, Foster PM. 2002. Fetal testosterone insufficiency and abnormal proliferation of Leydig cells and gonocytes in rats exposed to di(*n*-butyl) phthalate. *Reprod Toxicol* 16:19–28; doi:10.1016/S0890-6238(01)00201-5.
- Mylchreest E, Wallace DG, Cattley RC, Foster PM. 2000. Dose-dependent alterations in androgen-regulated male reproductive development in rats exposed to di(*n*-butyl) phthalate during late gestation. *Toxicol Sci* 55:143–151; doi:10.1093/toxsci/55.1.143.
- Parks LG, Ostby JS, Lambright CR, Abbott BD, Klinefelter GR, Barlow NJ, et al. 2000. The plasticizer diethylhexyl phthalate induces malformations by decreasing fetal testosterone synthesis during sexual differentiation in the male rat. *Toxicol Sci* 58:339–349; doi:10.1093/toxsci/58.2.339.
- Rudel RA, Camann DE, Spengler JD, Korn LR, Brody JG. 2003. Phthalates, alkylphenols, pesticides, polybrominated diphenyl ethers, and other endocrine-disrupting compounds in indoor air and dust. *Environ Sci Technol* 37:4543–4553; doi:10.1021/es0264596.
- Schettler T. 2006. Human exposure to phthalates via consumer products. *Int J Androl* 29:134–139; doi:10.1111/j.1365-2605.2005.00567.x.
- Sharpe RM. 2006. Pathways of endocrine disruption during male sexual differentiation and masculinisation. *Best Pract Res Clin Endocrinol Metab* 20:91–110; doi:10.1016/j.beem.2005.09.005.
- Sharpe RM, Skakkebaek NE. 2003. Male reproductive disorders and the role of endocrine disruption: advances in understanding and identification of areas for future research. *Pure Appl Chem* 75:2023–2038.
- Shultz VD, Phillips S, Sar M, Foster PM, Gaido KW. 2001. Altered gene profiles in fetal rat testes after in utero exposure to di(*n*-butyl) phthalate. *Toxicol Sci* 64:233–242; doi:10.1093/toxsci/64.2.233.
- Silva MJ, Barr DB, Reidy JA, Malek NA, Hodge CC, Caudill SP, et al. 2004. Urinary levels of seven phthalate metabolites in the U.S. population from the National Health and Nutrition Examination Survey (NHANES) 1999–2000. *Environ Health Perspect* 112:331–338; doi:10.1289/ehp.6723.
- Skakkebaek NE, Rajpert-De Meyts E, Main KM. 2001. Testicular dysgenesis syndrome: an increasingly common developmental disorder with environmental aspects. *Hum Reprod* 16:972–978; doi:10.1093/humrep/16.5.972.
- Swan SH, Main KM, Liu F, Stewart SL, Kruse RL, Calafat AM, et al. 2005. Decrease in anogenital distance among male infants with prenatal phthalate exposure. *Environ Health Perspect* 113:1056–1061; doi:10.1289/ehp.8100.
- Toppari J, Larsen JC, Christiansen P, Giwercman A, Grandjean P, Guillelte LJ Jr, et al. 1996. Male reproductive health and environmental xenoestrogens. *Environ Health Perspect* 104(suppl 4):741–803; doi:10.2307/3432709.
- Wilson VS, Lambright C, Furr J, Ostby J, Wood C, Held G, et al. 2004. Phthalate ester-induced gubernacular lesions are associated with reduced *insl3* gene expression in the fetal rat testis. *Toxicol Lett* 146(3):207–215; doi:10.1016/j.toxlet.2003.09.012.
- Zhang Y, Jiang X, Chen B. 2004. Reproductive and developmental toxicity in F1 Sprague-Dawley male rats exposed to di-*n*-butyl phthalate in utero and during lactation and determination of its NOAEL. *Reprod Toxicol* 18(5):669–676; doi:10.1016/j.reprotox.2004.04.009.